organic soil horizons and from 93 to 213 Mg·ha^{-1} for the underlying mineral soil. These values are higher than those estimated by Nave et al. (2010) for temperate forests (5–50 Mg·ha^{-1} organic horizons; 5–80 Mg·ha^{-1} for mineral soil). A recent review (Thiffault et al. 2011) comparing whole-tree and stem-only harvest impact in boreal and temperate forests in Canada concluded that C-poor soils (small pool size) were most sensitive to whole-tree harvest. However, when pool size is small (temperate and some boreal forest floor C stocks), the loss of a small amount of C can translate into a high proportional loss (Nave et al. 2010). Applying such high proportional losses to the boreal zone where forest floor C stocks are mostly large could lead to the potentially erroneous conclusion that harvesting in the boreal results in large losses of soil C to the atmosphere.

The high proportional loss of soil C in response to harvesting estimated for temperate systems may not occur in some of the dominant forest types of the Canadian boreal zone. This is especially true for black spruce forests, the most common coniferous forest cover type in the boreal forest (Lavoie et al. 2005). The dominant natural disturbance type in black spruce systems is wildfire, and consensus is emerging that harvesting has a less negative impact on the C budget of black spruce forests compared with wildfire (Bergeron et al. 2008). This is primarily because harvesting practices in most black spruce forests are less disruptive than wildfire (Amiro et al. 2001*a*), as harvesting is often restricted to the winter when the ground is frozen and covered in snow, thus, avoiding large disturbance to the soil (Lavoie et al. 2005). Studies examining black spruce stands in Ontario and Quebec generally indicated no response of mineral soil C to harvest disturbance. In some cases, reduction in forest floor C in younger stands was attributed to a change in harvest practices during the past several decades from horses to more disruptive mechanical logging (Brumelis and Carleton 1988) or post-harvest burns (Scheuner et al. 2004) rather than increase in decomposition rates. In black spruce systems conducive to paludification (a shift from non-peatland to peatland caused mainly by a change in the hydrologic balance to wetter conditions) (Fenton et al. 2010), low-impact harvesting can create conditions that favour C accumulation rather than loss (Lavoie et al. 2005).

Two emerging themes in temperate forest soil C research that may affect this conclusion are (*i*) most studies to date focus on C stock changes in the surface soil and do not account for the response of soil C at depth (to 100 cm or greater) and (*ii*) the apparent stability of mineral soil C may change in response to change in the environment, which can occur because of harvesting (Harrison et al. 2011). However, no research, to our knowledge, has been conducted in the Canadian boreal forest to study C dynamics at depth in response to harvesting or distinguish C that has accumulated from that which is stabilized (Jandl et al. 2007). In particular, we know little of the degree of stabilization of C (Marschner et al. 2008) in Canadian boreal forest soils (Norris et al. 2011).

3.3.6. *Invasive earthworms*

Since the time of Darwin (1881) it has been known that earthworms are important agents of soil formation and nutrient dynamics. However, the Canadian boreal forest zone has evolved in the absence of this significant ecosystem engineer (Evers et al. 2012). Non-native, primarily European, earthworms are being introduced to areas of boreal forest following the regional extinction of native species during the last ice age (Addison 2009). The primary vectors of spread are associated with human recreational and resource development activities (Cameron et al. 2007), the rate of which are expected to increase in the coming decades. Earthworms can have large impacts on GHG emissions (Lubbers et al. 2013) and DOM and soil C dynamics (Langor et al., Manuscript in preparation). They can reduce forest floor C stocks either through an increase in decomposition rates or transfer rates to the mineral soil (Langmaid 1964; Hale et al. 2005) and can stabilize C in the mineral soil (Shaw and Pawluk 1986). Reduction in forest floor C stocks by earthworms has implications for estimation of C emissions from fire that originate mainly from the combustion of the forest floor (Letang and de Groot 2012).

Although a model of the effects of earthworms on soil C was recently developed for temperate forests (Huang et al. 2010), it has not been adapted or tested for the boreal zone. Further to this, effects of earthworms are currently not included in landscape-scale models of boreal forest C dynamics because of insufficient data on the spatial distribution and rates of spread of earthworms in the boreal zone of Canada and because of an inadequate understanding of their effects on net C fluxes in boreal forests. However, given their site-level effects on forest floor and mineral soil C dynamics and their expansion in many parts of the boreal zone, the omissions of earthworm impacts in national-scale analyses of forest C budgets could contribute significant uncertainties to present and future estimates.

4. Future projections of Canada's boreal forest C cycle

The future C balance of the Canadian boreal forest will affect the global atmospheric C budget and influence the level of global mitigation efforts required to attain atmospheric CO_2 stabilization targets (Allen et al. 2009). This goal will be easier if forests continue to globally act as C sinks (Pan et al. 2011*a*). However, the potential for large increases in emissions from boreal forests and other terrestrial systems is real (e.g., Lavoie et al. 2005; Boisvenue and Running 2010; Metsaranta et al. 2011; Schuur and Abbott 2011) and would contribute to increases in atmospheric CO_2. Human-induced changes to the global environment have already affected forest systems (Boisvenue and Running 2006; Kurz et al. 2008*c*; Allen et al. 2010; Hember et al. 2012), and environmental changes are projected to intensify (IPCC 2007). The complexity and diversity of ecosystems combined with the range of environmental changes will result in regions and time periods with positive and negative feedbacks (i.e., net sources or net sinks) (Le Quéré et al. 2009; Boisvenue and Running 2010) and potentially large changes in the net balance over time (Morales et al. 2007; Xiao et al. 2010; Metsaranta et al. 2010). This section addresses future changes in the key drivers affecting the C balance in Canada's boreal forest, which include forest responses to environmental changes, changes in disturbance regimes, and changes in human activities and land use. For further review of future changes in the boreal zone, see Price et al. (2013).

4.1. Changes in forest dynamics

Productivity increases are already reported owing to warmer temperatures and longer growing seasons, increased atmospheric CO_2 concentrations, and N in temperate (Hember et al. 2012) and boreal forests (Magnani et al. 2007; Briffa et al. 2008; Hickler et al. 2008). A recent review of measured changes in forest productivity globally found that 75% of papers reviewed reported increased productivity and the remaining studies declining (10%), mixed (8%), or no trend (4%) in productivity (Boisvenue and Running 2006). Other reports of changes already occurring were outlined in section 3.3.1, and several process modelling and experimental studies, some conducted in Canada's boreal forest, project future changes to growth rates (Chen et al. 2000; Gamache and Payette 2004; Hickler et al. 2008; Silva et al. 2010; Zhao and Running 2010; Beck et al. 2011; Berner et al. 2011; Toledo et al. 2011).

Factors that could potentially limit the growth response include moisture constraints, nutrient availability, thin soil cover, and (or) pest disturbances (Jarvis and Linder 2000; Lafleur et al. 2010; Beck et al. 2011). Responses to environmental change will vary by species (Cole et al. 2010), ecological region (Paquette and Messier 2011), provenance (McLane et al. 2011*a*), and management regime

(Cyr et al. 2009). Drought (Allen et al. 2009; van Mantgem et al. 2009; Zhao and Running 2010), disease (Sturrock et al. 2011), insects (Hicke et al. 2012) or genetic adaptation (McLane et al. 2011*b*) may decrease future productivity or increase future mortality (Peng et al. 2011; Ma et al. 2012) or both. Projected changes in disturbances patterns that may affect the C balance of boreal forest are discussed in a subsequent section.

Many C budget projections for Canada's boreal forest assume that the distribution of tree species does not change over time (e.g., B. Smith et al. 2011). Climatically suitable ranges for tree species will likely exhibit large-scale redistribution over the 21st century (e.g., Hamann and Wang 2006; McKenney et al. 2007; Schneider et al. 2009). Species ability to shift their distribution to spatially track changes in climate will depend on dispersal ability (Greene et al. 1999; Nathan et al. 2011), competition (Loehle 1996), and soil conditions (Lafleur et al. 2010), as well as interactions among these factors (Leithead et al. 2010) and with disturbances (Greene and Johnson 2000; Greene et al. 2004; Simard and Payette 2005; Johnstone and Chapin 2006*a*, 2006*b*) and regeneration success (Classen et al. 2010). The future distributions of tree species are highly uncertain and depend strongly on the assumptions of the model used to make the projection (Loehle 2000; McKenney et al. 2007; Morin and Thuiller 2009; Mbogga et al. 2010). It is unlikely that species will be able to migrate as quickly as their climatically suitable ranges change (Aitken et al. 2008), resulting in local populations that are progressively more genetically maladapted to their new climate. If unable to migrate, species will only persist locally through genetic adaptation to new conditions or face extirpation (Aitken et al. 2008; Barnes 2009). Inter species ability to adapt also differs (Trindade et al. 2011). Over the 21st century, the response of trees to shifts in climatic niches will likely be a combination of gradual change in areas where seed dispersal limits distributions and rapid shifts to new ecosystem states where thresholds are surpassed (Chapin et al. 2004). Along with maladaptation and mortality or a gradual or sporadic change in species will come a change in productivity and, hence, a change in C balance.

Carbon stocks may also be influenced indirectly by changes in the distribution of boreal forest relative to other ecosystem types (e.g., Beck et al. 2011). The southern boundary of the western Canadian boreal forest is a forest–grassland ecotone. As a result, there is a higher risk that these transitions will result in a shift to nonforest communities, either grassland typical of the prairies or sparse woody vegetation currently typical of the aspen parkland (Hogg and Hurdle 1995; DeSantis et al. 2011). In northern Quebec and other parts of the Canadian boreal zone, repeated fires can lead to a transition from closed-crown forests to lichen woodlands with possible reduction (Girard et al. 2008) or possible increases (Lavoie et al. 2005) in C stocks but also altered albedo (Bernier et al. 2011). The southern boundary of the eastern Canadian boreal forest is typically a boreal–temperate forest ecotone (i.e., hemiboreal subzone, Brandt 2009), where transitions are likely to result in a shift in dominance from boreal to temperate tree species representative of Great Lakes St. Lawrence or tolerant hardwood forests, rather than a loss of forest cover (Leithead et al. 2010). The influence of these effects on C dynamics is still uncertain. A transition to grassland would result in a reduction in C stocks; but a change in tree species could affect C stocks in either direction, depending on the new species and soil C dynamics.

Climate warming can also lead to changes in the dynamics of vegetation at treeline (Körner 2012). Increased tree recruitment at the forest–tundra ecotone can result in an advance of both altitudinal and latitudinal treeline into areas of tundra (e.g., Gamache and Payette 2005). A recent global review shows that about half of the studies examining changes over the last century have recorded an advancing treeline (Harsch et al. 2009). Northern forest extension could lag warming for a few decades and transient species associations may initially develop, but over time forest limits could advance to those experienced before the Little Ice Age (MacDonald et al. 2008). Climate warming can also lead to increases in vegetation density, particularly shrub vegetation, in the northern boreal forest and the Arctic, which has been demonstrated by remote-sensing studies (e.g., Sturm et al. 2001; Pouliot et al 2009; Fraser et al 2011; McManus et al. 2012), also referred to as vegetation greening. Figure 10 shows an example of this process (NASA 2012*a*). A thorough review of the processes of shrub vegetation thickening and advance, including both promoting and limiting factors, is provided by Myers-Smith et al. (2011). The total impact of treeline advance and vegetation greening in the Arctic on future boreal forest C dynamics is difficult to predict, but their main impact will be on the relative distribution and extent of forest and tundra vegetation. There are also complex interactions between factors such as local environmental conditions (e.g., Mamet and Kershaw 2013), fire disturbance (e.g., Arseneault and Payette 1992), and their interactions (e.g., Munier et al. 2010) that still remain to be completely understood.

Process modelling of environmental impacts on growth rates needs to be linked to modelling of ecosystem and species shifts across ecologically complex regions to help assess the effects of opposing factors in forest dynamics and the C balance of these forests. Modelling capabilities are improving but are not yet reaching the ability to estimate future forest productivity. There is presently no agreement on the direction, magnitude, or cause of net changes in the future productivity of Canada's boreal forest (e.g., Bernier et al. 2010; Shanin et al. 2010).

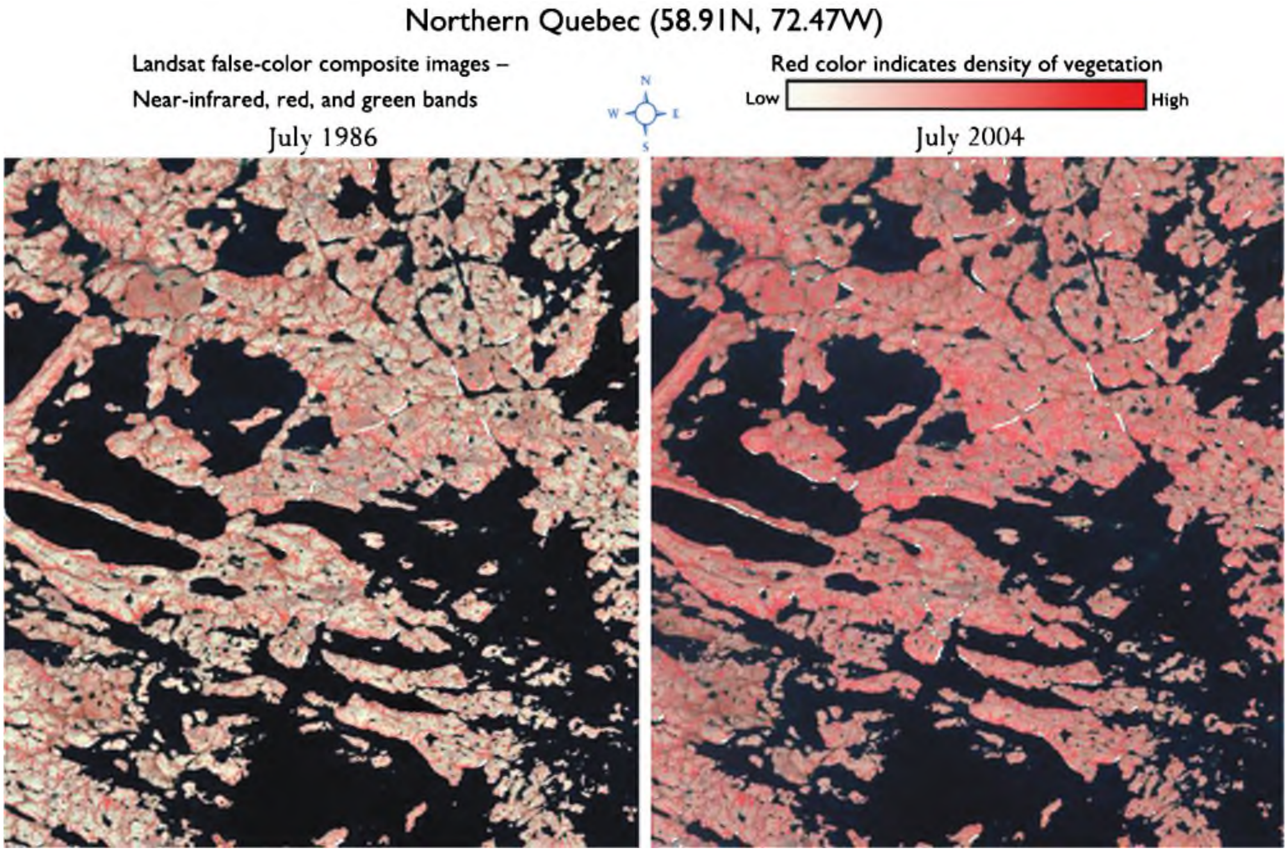
4.2. Changes in decomposition rates

Greenhouse gas emissions to the atmosphere from boreal soil C are expected to increase with future warming because of the temperature sensitivity of soil organic matter decomposition that is commonly used in models (Rodrigo et al. 1997; Peltoniemi et al. 2007) and the presumption that soil temperature will increase apace with air temperature. Predicted higher air temperatures are also expected to affect NPP and so it is the difference in the rates of changes in NPP and R_h that will determine the change in NEP.

The degree to which soil decomposition is sensitive to temperature remains uncertain, with some studies suggesting higher sensitivity (Knorr et al. 2005) and others lower (Davidson et al. 2000; Giardina and Ryan 2000) sensitivity to temperature increase. Experiments have found a range in temperature sensitivity with Q_{10} values from 1 to 4 (Boone et al. 1998; Irvine et al. 2005; Parè et al. 2006; Gaumont-Guay et al. 2008; Fissore et al. 2009). Conflicting or unexpected responses of the temperature sensitivity of R_h may be observed because studies measuring and modelling decomposition do not directly represent the soil (e.g., microbial activity, stabilization with inorganic components, and aeration) and landscape (e.g., permafrost thaw, thermokarst, change in insulation from peat, forest or snow cover, and hydrology) processes controlling R_h. However, these processes may have counteracting effects (Davidson and Janssens 2006; Conant et al. 2011) and respond differently to temperature change. Recognition that modelling of soil C dynamics may be more complex than previously thought (Dungait et al. 2012) has led soil C modelling researchers to propose changes to the fundamental approach to modelling soil C and its response to climate change (Wutzler and Reichstein 2008; Allison et al. 2010; Conant et al. 2011; Schmidt et al. 2011). Suggestions include modelling of soil C should abandon the structure of pools with intrinsic decomposition rates, including the vaguely defined "recalcitrant" fraction, and should instead move towards the representation of processes directly controlling R_h (Conant et al. 2011; Schmidt et al. 2011).

Aside from uncertainties associated with the sensitivity of decomposition to temperature, we also need to understand the magnitude and trajectory for boreal soil temperature (or soil climate) change in response to climate change. Long-term change in soil

Fig. 10. A comparison of area in northern Quebec showing increased vegetation density between 1986 and 2004 (credit: Jeff Masek). Available from http://www.nasa.gov/topics/earth/features/shrub-spread.html [accessed 10 September 2012].



temperature is poorly documented but studies in Canadian (Zhang et al. 2005) and European boreal zones (Helama et al. 2011) have shown that it will not necessarily exhibit the same trends in air temperature as is often presumed in C models. The discrepancy between air and soil temperature and their shifts with climate change in the boreal is likely attributable to spatially and temporally complex interactions between edaphic and landscape factors affecting hydrology, permafrost, snow cover, peat, and vegetative cover (Bond-Lamberty et al. 2007*a*; O'Connor et al. 2010; Hennon et al. 2012). The response of these factors to climate change affects the trajectory and rate of change in soil climate (including temperature, moisture, and aeration — oxic versus anoxic condition) for decomposition.

Permafrost thaw is a process of great concern in the discontinuous permafrost zone of the boreal zone (Fig. 3) because it has the potential to significantly contribute to atmospheric feedbacks and climate change (Schuur and Abbott 2011). It not only changes soil climate for decomposition but also the size of the C pool available for decomposition because the C is no longer frozen or permafrost thaw has altered the water table and the proportions of C under oxic and anoxic conditions. The movement of C from frozen to unfrozen state may proceed at rates an order of magnitude higher than the direct effect of temperature sensitivity (Schuur et al. 2009). Once thawed, the rate of permafrost C decomposition may be very high in the cases where C stability was low at time of freezing (Zimov et al. 2006) or relatively low for some peatland types if the organic matter was already relatively decomposed at the time of freezing (O'Connor et al. 2010). Recent studies have suggested that, during the initial phases of permafrost thaw, uptake of C resulting from increases in plant productivity can initially compensate for loss of C due to increases in R_h, but that a tipping point will be reached where R_h exceeds NPP and that the resulting C emissions are expected to continue into the future (O'Donnell et al. 2012; Schaphoff et al. 2013). Tarnocai (2006) rated the sensitivity of peatlands to climate change into six classes ranging from "no change" to "extremely severe". We estimate that frozen peatlands in the severe and extremely severe sensitivity classes contain 34 Pg C in the boreal zone (Table 1). Tarnocai (2006) also estimated that 36 Pg C in unfrozen peatland is at equal risk and some portion of these peatlands also occurs in the discontinuous permafrost zone. However, it is also this zone that presents the most challenge to models used to predict the response of permafrost thaw to climate change (Schaefer et al. 2011) because of the models' limited ability to represent small-scale heterogeneity and feedbacks that can lead to both positive and negative effects on permafrost stability (Strack et al. 2008; Grosse et al. 2011; Schaefer et al. 2011). Schaefer et al. (2011) reported on model estimates for permafrost degradation in the 21st century that range from 16% to 85% with their own model predicting a 29%–59% reduction. Caution must be exercised, however, when applying these rates to the Canadian boreal zone because they are largely derived from permafrost areas outside of the Canadian boreal zone (global arctic and subarctic zones) and have the common challenge of representing the heterogeneity of the discontinuous permafrost zone that dominates the boreal zone. Although it is generally accepted that permafrost and peatland soils in the Canadian boreal zone will be sensitive to climate change (Tarnocai 2006; Zhang et al. 2008), it is still unclear if the outcome will be a net increase in productivity or emissions to the atmosphere as CO_2 and (or) methane (CH_4) (Lavoie et al. 2005; Turetsky et al. 2007; Yu et al. 2011). Uncertainties are particularly high for the prediction of CH_4 emissions from peatland and permafrost thaw in response to climate change (O'Connor et al. 2010), and some of the processes important to the cycling of methane are just being dis-

covered. For example, recent research (Kip et al. 2010) found that CH_4 released from the decay of sphagnum mosses can be oxidized by symbiotic methanotrophs and the C reassimilated by the moss when submerged, which provides a mechanism to potentially reduce CH_4 emissions.

In forested nonpermafrost zones, increases in NPP with climate change can result in higher inputs of C to soil from foliage and fine root turnover (Matamala et al. 2003; Iversen et al 2008), but this does not necessarily result in an increase in soil C (Schlesinger and Lichter 2001). Tree species have a strong influence on root allocation responses to CO_2 increase (Matamala et al. 2003), and whether or not R_h is stimulated by increased inputs depends on soil characteristics (Bader and Körner 2010). Priming of soil respiration by greater inputs of root and foliar C may result in greater respiration of older soil C (Trueman and Gonzalez-Meler 2005; Fontaine et al 2007). Understanding and modelling landscape-level hydrology is critical to predicting the C budget of the most northerly unmanaged forest area (see section 3.2.2) and also important in the more southerly managed forest area that may be unaffected by permafrost but where topographic controls on drainage patterns influence soil C stocks (Webster et al. 2011) and soil respiration (Webster et al. 2008).

4.3. Future disturbances

Carbon dynamics in the Canadian boreal forest have historically been dominated by natural disturbances, with fire as the dominant disturbance in the western boreal forest (Fig. 4; Bergeron et al. 2004; Lavoie et al. 2005; Balshi et al. 2007) and both fire and insects affecting the eastern boreal forest (Blais 1983). Recent changes in climatic conditions have contributed to increased impacts of drought and insects (Peng et al. 2011). Positive feedback to climate change could result from increasing disturbance frequency and intensities. This section reviews some of the potential future effects of these disturbances on the C balance of the boreal forest. For additional information, see also Price et al. (2013).

Projected future weather conditions in the Canadian boreal forests increase the probability of fire occurrence (and hence of area burned) over the 21st century (Flannigan et al. 2005*b*, 2009; Balshi et al. 2009; Krawchuck et al. 2009; Hély et al. 2010). Increases in fire and other disturbances will contribute to increased emissions and forests will (other things being equal) store less C, thus contributing towards "positive feedback" to climate change (Metsaranta et al. 2010; Melillo et al. 2011). However, these effects are not likely to occur uniformly. Area burned in Canada's boreal forest fluctuates widely from year to year (Armstrong 1999; Amiro et al. 2001*b*; Stocks et al. 2003). As a result, detecting changes in fire regimes from short time series of data are almost impossible (Metsaranta 2010). In addition, much of the cumulative area burned over a given period of record occurs in a small number of years with large area burned, and the frequency and magnitude of these extreme fire years is also highly uncertain (Metsaranta 2010). Vegetation succession that increases the proportion of deciduous forests in the boreal forest region could provide a negative feedback with respect to the projected increase in fire (Johnstone et al. 2011). However, quantitative studies on the selectivity of burning behaviour with respect to forest type in Canadian boreal forests are inconsistent. Cumming (2001) supports the hypothesis that coniferous forests burn more than their proportional contribution to landscape composition, but Podur and Martell (2009) suggest that all forest types burn in proportion to their composition.

Furthermore, timing of the fire has been shown to have an effect on the depth of burn, with late-season burns resulting in more of the ground surface organic matter consumed (Turetsky et al. 2011), which has significant consequences on the C balance. Such conditions could also allow extreme fire events with prolonged smoldering phases (and the associated high CH_4 emissions) under snow and during winter months.

Impacts are expected to be greater in the drier, continental western boreal than the eastern Canadian boreal forest (Flannigan et al. 2005*b*; Balshi et al. 2009), and increased fire occurrence in the future is projected to overwhelm the capacity of fire-management agencies to mitigate these effects through increased suppression effort (Flannigan et al. 2005*a*; Podur and Wotton 2010). Projections of the impact of increases in area burned in all of Canada's managed forest over the 21st century, assuming that area burned would increase by a factor of 4 in most of western Canada and a factor of 2 in eastern Canada, suggest that to maintain ecosystem C stocks NEP would have to increase by about 25% to offset increased fire emissions. An increase in NEP of this magnitude, sustained over time and over all areas in which disturbances increase, is not likely (Kurz et al. 2008*a*; Metsaranta et al. 2010).

Climate change is predicted to affect future insect disturbances in several ways: range expansions northward and to higher elevations (Régnière et al. 2010; Safranyik et al. 2010); increased temperature allowing insects to mature more quickly, reduce winter mortality, and increase summer productivity (Raffa et al. 2008); and changes in the synchrony of insect life cycle stages and plant phenology (Nealis and Régnière 2004; Régnière et al. 2009). The net impacts of these changes on the forest C balance are difficult to predict, but insect outbreaks can have large impacts on C stocks and fluxes (Kurz et al. 2008*c*; Dymond et al. 2010; Hicke et al. 2012). The main impacts of insects in the boreal forests of Canada have historically been confined to the southern regions, and the potential for range expansion into regions where host tree species are present but have historically not been challenged by insects could result in increased tree mortality and greater reduction in C stocks. Examples include forest tent caterpillar in the Northwest Territories in 1995 and 1996 (Brandt 1997) and the potential spread of mountain pine beetle on jack pine across Canada's boreal zone (Safranyik et al. 2010).

Parts of the boreal forest, particularly in western Canada (Michaelian et al. 2011), but also in parts of eastern Canada (Hély et al. 2010), are expected to experience more frequent and severe droughts in the 21st century, potentially impacting several ecosystem processes that influence forest C dynamics (van der Molen et al. 2011). Increases in drought-induced forest mortality have already been observed globally (Allen et al. 2010; van Mantgem et al. 2009; Huang and Anderegg 2012) and in Canada (Michaelian et al. 2011; Peng et al. 2011; Ma et al. 2012). Precipitation is found to influence forest productivity in both tree-ring (Beck et al. 2011) and flux tower (Schwalm et al. 2010) studies. In addition, drought can influence soil C dynamics, with dry conditions potentially resulting in reduced decomposition rates (Allison and Treseder 2008; Smyth et al. 2010) that under some conditions can offset C balance impacts resulting from productivity losses. The net impact of these effects on the C balance of boreal forests in Canada has not yet been quantified. Most likely, impacts will vary by forest types and regions depending on moisture regimes and site conditions, as well as interactions with insects and pathogens. Deciphering the physiological mechanism by which trees decline and die under drought will soon contribute to better modelling and prediction of drought events and their effects on C balance (Anderegg et al. 2012).

4.4. Land-use change

Future economic, social, and climatic conditions will affect the future rates of deforestation across the boreal zone. In the northern boreal forest, large individual events and developments are anticipated to have the main impact. For example, several hydroelectric developments are being considered over the next 25 years in northern Manitoba, Quebec, Labrador, and British

Columbia. Construction of new mines and their access roads, programs to connect northern communities to the electric grid and permanent roads, and other efforts to improve access to northern resources are all expected to contribute areas of deforestation.

The main forestry access road system in the boreal forest is becoming largely developed but is expected to continue to move northward into previously unaccessed areas of commercially harvestable forest. The oil and gas infrastructure is also expected to expand where resources have been found. In the oil sands regions, increased use of subsurface steam-assisted gravity drainage (SAGD) methods for oil extraction will contribute deforestation from well pads, pipeline, access roads, and steam generating and processing facilities. Activity in the Northwest Territories is also increasing, and the construction of the Mackenzie Valley pipeline will result in areas of deforestation. Within the mineable oil sands district, considering the total mineable area, its forest cover, planned developments, and expert opinion on future development, a reasonable estimate of the total area eventually disturbed is in the order of 200 kha of which approximately 75%–85% will involve forest.

The decline in deforestation for agricultural land conversion is expected to continue in the future, although changes in economies, demand for agriculture-based biofuels, and government policy under climate change could also result in agricultural expansion into forest regions. Most land currently under agriculture in the boreal zone is capable of being afforested. Future afforestation rates are expected to be low and similar to current rates, in the absence of government incentive programs (Lemprière et al. 2013). In the boreal Clay Belt regions of Quebec and Ontario, considerable abandonment of agriculture land has occurred over the past decades. Some of this will eventually revert to forest; and where these forest areas are included in provincial inventories, the C consequences are captured in the regional C balance estimates. It is unknown whether abandonment of agriculture land in the Clay Belt will continue in the future or whether this land will be reclaimed for agricultural uses, but growing world population, higher demand for food, and raising food prices all increase the pressure to reclaim land for agricultural uses.

4.5. Net carbon balance

The future net balance of C emissions and removals in Canada's boreal forest will be affected primarily by changes in forest productivity (section 4.1), decomposition rates (section 4.2), and natural disturbances (section 4.3). While it remains impossible to predict with certainty the resulting net C balance under future climate conditions, the "asymmetry of risk" (Kurz et al. 1995) is of concern: for boreal forests stands to reach maturity and maximum C storage, many decades of survivable growing conditions must prevail, but it takes only a single extreme event such as drought, windthrow, fire, insects, or other disturbances to kill trees or stands. With climate change predictions including more frequent extreme climatic events (Rahmstorf and Coumou 2011; Hansen et al. 2012), increases in natural disturbances (Flannigan et al. 2005*b*; Balshi et al. 2009), and maladaptation of forest ecosystems to shifting climate conditions (Aitken et al. 2008; Barnes 2009), the probability that boreal forests C stocks will increase under climate change scenarios is lower than the probability that they will decrease.

A second argument for the likely decrease in boreal forest C stocks with climate change is that considerably more C is stored in DOM and soil C pools than in biomass, largely because cold, wet, anoxic, and frozen environments have delayed or prevented decomposition. As discussed previously, warming is generally predicted to increase decomposition, leading to reductions in DOM and soil C pools that are likely greater than a possible increase in biomass C pools, resulting in an anticipated net decrease in boreal forest C stocks.

5. Knowledge gaps and monitoring needs

The net C balance of Canada's boreal forest is dominated by two large fluxes: NPP and R_h, processes that continuously occur in all forest ecosystems. Any changes in environmental conditions such as climate change, CO_2 fertilization, and N deposition that affect NPP and R_h have the potential to cause large changes in the net C balance, in particular if environmental changes have opposing impacts on the two fluxes, e.g., decreasing productivity and increasing respiration losses. In addition to the continuous processes of growth and decomposition, some ecosystems are in some years affected by anthropogenic and natural disturbances, and the boreal forest C balance will be strongly affected by changes in disturbance regimes, i.e., the frequency, intensity, and types of disturbances.

To reduce the uncertainties in the estimates of the current and future C balances of the boreal forest, research needs to be directed to improve understanding of (*i*) continuous processes determining NPP and R_h, (*ii*) disturbance-related processes, (*iii*) interactions between disturbance and ecosystem production, and (*iv*) interactions among landscape distribution of forests, environmental drivers of disturbance, and successional trajectories.

Models of forest C dynamics have been used successfully to derive estimates of net C balances that take into account the broad distribution of forest characteristics including species, forest age, site conditions, and the impacts of natural disturbances and forest management. By necessity, such models incorporate assumptions about homogenous conditions within forest stands and landscapes. Although the spatial resolution of models used at the scale of the boreal forest has increased by more than three orders of magnitude over the past 20 years, the "average" stand represented by such national-scale models today is typically about 100–1000 ha. Depending on vegetation characteristics, topography, and soil conditions within average stands of such size, the ecological processes that determine C fluxes can be occurring at a range of rates and respond differentially to environmental changes. Further improvements in (*i*) the spatial resolution of modelling approaches down to 1 ha resolution and (*ii*) the availability of spatially-explicit data on forest characteristics, topography, and soils at the increased resolution have the potential to contribute to reducing uncertainties of C stock and C flux estimates, provided that sufficient data on environmental characteristics are available at that fine spatial scale (Canadian Forest Service, Natural Resources Canada 2012). Many forest ecosystems models operate on annual time scales, and increasing the temporal resolution would allow the improved representation of processes occurring at seasonal, monthly or daily time scales. Efforts to reduce uncertainties in C budgets by increasing spatial and temporal resolution of models will substantially increase the demands for input data and computing resources.

Improvements in remote-sensing techniques combined with forest ecosystem models will likely achieve further reductions in the uncertainty of disturbance-related C flux changes in the coming years. In contrast, reducing uncertainties about subtle changes in fluxes in response to fluctuating environmental conditions will remain an ongoing challenge. Every 1 g m^{-2} $year^{-1}$ change in net fluxes over Canada's boreal forest sums to a change of 2.7 Tg $year^{-1}$ (or nearly 10×10^6 Mg $year^{-1}$ of CO_2) in the boreal forest C balance; thus, even subtle changes in fluxes, undetectable with currently available methods, occurring in synchronicity over large areas can have large impacts on the global C cycle. Eddy covariance flux towers have successfully been used to quantify high-frequency flux responses to environmental drivers, but measurements have been limited to a small number of sites, each with a small footprint and relatively short observation period. Thus, spatial and temporal upscaling of ecological processes remains a major challenge.

New techniques to quantify changes in growth and mortality rates in response to environmental change using tree-ring data

(Metsaranta and Kurz 2012) and permanent sample plots (Hember et al. 2012) offer opportunities to gather empirical data on forest responses to climate change over much larger areas and multi-decadal time periods. Such data can help inform and constrain process models. Combined with site-specific ecosystem process models, e.g., ecosys (Grant et al. 2007), InTec (Chen et al. 2000), or 3PG (Landsberg and Waring 1997), these data offer a path to further reducing uncertainties in carbon flux estimates (Keenan et al. 2012). However, as climate change continues to affect Canada's boreal forests, ongoing monitoring of forest growth and mortality responses to climate change will be required. In recent years, the numbers of climate-monitoring stations, permanent sample plots, and flux towers in Canada's boreal forest have all decreased while the need for monitoring data has increased, and the ability to extract scientifically relevant knowledge from such measurements has improved.

Traditional forest inventories, permanent sample plots, tree-ring analyses, and meta-analyses provide significant insights into forest responses to environmental changes. But, in boreal forests, a larger proportion C is stored in DOM and soil C pools for which much fewer measurements exist. Moreover, considerable uncertainty remains on the impacts of climate change on R_h (Pendall et al. 2004; section 4.2). Research needs to reduce this knowledge gap include linking different agents of tree mortality to fall and decay rates, time series of soil C stock measurements using consistent methodologies to enable the detection of trends in C stocks in response to environmental changes, monitoring of changes in permafrost distribution and active layer depth, soil-warming experiments to better understand processes that will change in the future, transect studies along climate gradients, and the quantification of soil C dynamics.

Effects of increased atmospheric CO_2 on aboveground production have been investigated, but effects on belowground processes have received much less attention and as a result are not well understood (Iversen et al. 2008). Carbon dioxide fertilization responses could cause changes in deep soil C pools, for example, through changes in C allocation to fine roots, through increased production, and through allocation of fine roots to deeper layers in the soil profile. Therefore, plot data on fine root production, its vertical distribution in the soil profile, and data on turnover rates are needed to help quantify the effects of increased atmospheric CO_2 concentrations and climate change (e.g., Olesinski et al. 2012).

Gaps also remain in the representation and quantification of processes and pools. The contributions to Canada's boreal C cycle of bryophytes, deep organic soils, and permafrost thawing are three examples of areas in which insufficient understanding and quantification at the national scale contribute significant uncertainties to the estimates of pools, their current C balance, and their projected future changes.

Reduction in uncertainties of regional- and national-scale estimates will require models that integrate environmental and climate data, forest inventories, and information obtained from remote sensing to scale up site-specific knowledge to larger areas and over longer time periods. Data assimilation approaches that combine ecosystem models and empirical data and constrain flux estimates using plot-level data, EC tower data, remote-sensing information, and inverse-modelling approaches offer new methods to reducing uncertainties (Richardson et al. 2010*a*; Chen et al. 2011). Remote-sensing techniques are increasingly detecting large-scale changes in forest reflectance properties that are correlated with forest productivity (Zhao and Running 2010; Beck and Goetz 2011). While the interpretation of such observations remains under development, opportunities exist to improve the scientific understanding of remotely sensed responses through comparisons against ground observations, including permanent sample plot data, tree-ring measurements, and flux tower measurements, as discussed previously.

Geological and edaphic conditions may be extremely variable even at very fine spatial scales, yet these have tremendous impact on C cycling (Ju et al. 2006). In particular, soil water and drainage needs to be better known to estimate the C balance of high latitude ecosystems (Ju et al. 2010). Predicting hydrological patterns is especially difficult in the unmanaged forest area because of the combined effects of geology, soils, permafrost dynamics, and peat. Peat exerts strong controls over hydrothermal regimes and water retention and, where it is thick, can obscure underlying geological material that in themselves can determine water retention and flow patterns (Holden 2005; Heinemeyer et al. 2010). Because geological landforms, composition, and soils differ significantly between the Boreal Plains in the west and the Boreal Shield in the east, their hydrological systems need to be modelled differently (Devito et al. 2005). This is especially important in the Boreal Plains where geological materials are compositionally complex (retain water), topography has low relief, and the landscape is a mosaic of peatland and upland areas with different hydrological properties that interact in complex ways (C. Qualizza et al., personal communication, 2012).

Reducing uncertainties about boreal forest C balances will require well-coordinated interdisciplinary research programs, national-scale data sets on forest conditions and forest change obtained from forest inventories and ongoing monitoring programs (including remote sensing), and ecosystem-modelling approaches supported by advanced computing infrastructure to synthesize and integrate the large volume of data that would be generated by such research programs (Canadian Forest Service, Natural Resources Canada 2012).

6. Conclusions

We estimate that, since 1990 Canada's managed boreal forest has acted as C sink of 28 Tg C year^{-1}, removing CO_2 from the atmosphere to replace the 17 Tg of C annually transferred out of the forest in timber-harvesting operations and store an additional 11 Tg of C in biomass, dead wood, litter, and soil C pools (Stinson et al. 2011). A large fraction (~57%) of the C harvested since 1990 remains stored in wood products in use and in solid waste disposal sites in Canada and abroad, replacing C emitted from the decay or burning or wood harvested prior to 1990 and contributing to net increases in HWP and landfill C storage pools. The use of these HWP products has contributed to reduced emissions in other sectors where HWP have replaced more emission-intensive products such as concrete, steel, and plastics; but the magnitude of the substitution benefits in Canada and abroad is subject to ongoing research.

Carbon balance estimates for the unmanaged boreal forest are currently limited to "poorly constrained" process models with high uncertainties owing to the lack of forest inventory data and limited understanding of the extent and impacts of permafrost thawing and climate change. In the unmanaged forest, fire is the predominant disturbance type, with very minor insect disturbances, no forest harvesting, and very small areas affected by land-use change. Thus, the unmanaged forest is likely to have been a sink in the second half of the 20th century (owing to low fire disturbances; Kurz and Apps 1999) and has recently transitioned to a smaller sink or a small source as the area annually burned has increased, but this conclusion is highly uncertain.

Biomass C stocks (an indicator that can be used as a proxy for growing stock volume) are increasing slightly in the managed boreal forest, suggesting that current rates of harvesting, natural disturbances, and forest management (e.g., fire suppression, planting, and other silvicultural activities) are sustainable with regard to biomass and total ecosystem C stocks. However, there are regional differences in harvest and disturbance rates, and there may be regions within the boreal zone where the combined impact of human and natural disturbances is currently larger

than the ability of these forests to sustain biomass stocks, in particular in areas with significant impacts on the forest from other industrial sectors. Moreover, sustainability of human actions is not assessed by C stock changes alone. Our conclusions are affected by numerous uncertainties, including uncertainties about rates of regeneration following disturbances and rates of growth, with possible errors in either over- or under-estimating rates of C stock changes.

The single largest threat to C stocks and future C balances in Canada's boreal forest is human-caused global climate change. Large C stocks have accumulated in the boreal zone because decomposition of organic matter is limited by cold temperatures and often anoxic environments. Increases in temperatures and disturbance rates could result in a large net C source during the remainder of this century and beyond. Uncertainties about the response to global change factors remain high, but we emphasize the asymmetry of risk that sustained large-scale increases in productivity and C uptake are unlikely to be of sufficient magnitude to offset higher C losses from increases in area burned and heterotrophic respiration (Kurz et al. 1995, 2008*a*; Metsaranta et al. 2011).

Reducing the uncertainties of the current and future C balance of Canada's 270 Mha of boreal forest requires addressing gaps in monitoring, observation, and quantification of forest C dynamics, with particular attention to Canada's 125 Mha unmanaged boreal forest with large areas of deep organic soils, peatlands, and permafrost containing large quantities of C that are vulnerable to the impacts of climate change.

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Appendix A: Land-use change data sources

Derivation of total area of land-use change in the boreal zone

Data used to estimate the reported 11.8 Mha of land conversion throughout Canada's boreal forest during 1990–2008 (see section 3.2.3.1) include a 1990 land-use map (Hill et al. 2011) derived from a rule-based conversion of several data sources (e.g., National Topographic Database and Earth Observation for Sustainable Development of Forests (Wulder et al. 2008) and Geocover (Koeln et al. 2000) land cover maps), a ca. 2010 crop map produced by Agriculture and Agri-Food Canada (T. Fisette, personal communication, 2011), and information on the land area flooded by hydroelectric reservoirs (e.g., Eichel and Leckie 2006; Lee et al. 2011) (see section 3.2.3.1).

Details of annual deforestation 1990–2008

Table A1 summarizes the estimated deforestation rate by industrial category from 1990 to 2008.

Methodology used for deforestation estimates

Deforestation estimates are derived from a national monitoring system operated by the Canadian Forest Service that generates national deforestation estimates for each year from 1970 to present (Leckie et al. 2006, 2009). It is based on interpretation of Landsat satellite images for three time periods (ca. 1975–1990, 1990–2000, and 2000–2008). Additional information is used such as historical aerial photographs; recent high-resolution satellite images; geospatial information such as forest inventory, hydroelectric development, and oil and gas infrastructure data; and sometimes verification through aerial observation or ground visits. Where forest cover loss is related to direct human-induced land clearing for nonforest land use, the area is recorded as deforested, predisturbance forest cover identified, and the sector or industrial category responsible for land clearing recorded. Mapping is conducted on a network of sample cells with a sampling intensity of 4%–12% of the area in the southern boreal forest. In the northern boreal forest where deforestation is extremely rare, mapping is conducted only around known high-activity areas and large events such as mines, hydroelectric developments, and transmission lines, which are easily detected on satellite images. The mapping in each time period is scaled according to sample intensity and interpolated over the three time periods to give annual deforestation estimates.

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Table A1 appears on the following page.

Table A1. Area (kha) in Canada's boreal zone annually deforested by industrial class responsible for deforestation (1990–2008).

Industrial category	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	Average per year	Sum
Agriculture	29.1	26.3	23.4	20.6	17.7	14.8	14.4	14.1	13.7	13.3	12.9	12.5	12.1	11.7	11.4	11.4	11.4	11.4	11.4	15.4	293.5
Hydro flooding																				—	
Flooded standing forest	—	—	—	34.2	0.7	—	—	—	—	—	—	—	8.7	—	—	0.5	27.6	—	—	3.8	71.7
Hydro reservoir	0.9	1.6	2.1	0.6	—	0.0	—	0.4	3.1	5.4	—	—	—	2.1	2.7	4.5	—	—	—	1.2	23.4
Forestry	2.4	2.4	2.5	2.5	2.5	2.6	2.7	2.7	2.8	2.9	3.0	3.1	3.1	3.2	3.3	3.3	3.3	3.3	3.3	2.9	54.9
Municipal	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.7	0.7	0.7	0.7	0.7	0.8	14.8
Oil and gas	5.3	5.4	5.5	5.6	5.7	5.8	6.3	6.9	7.4	7.9	8.4	9.0	9.4	10.0	10.5	10.5	10.5	10.5	10.5	7.9	150.8
Other																				—	
Hydro infrastructure	1.7	2.2	2.3	1.8	1.0	1.0	1.3	1.3	1.1	0.9	0.5	0.5	0.6	0.8	1.0	0.8	1.3	1.5	1.3	1.2	23.1
Industry	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	4.5
Mining	0.8	0.9	0.9	0.9	1.1	1.1	1.3	1.1	1.1	1.1	1.1	1.2	1.1	1.1	1.1	1.0	1.0	1.0	0.9	1.0	19.9
Peat mining	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8
Recreation	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.2	2.9
Transportation	0.9	0.9	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.6	0.7	0.7	0.6	0.7	14.1
Total	42.5	40.9	38.9	68.3	30.8	27.6	28.2	28.5	31.2	33.5	27.8	28.1	37.0	30.6	31.6	33.6	56.8	29.3	29.0	35.5	674.2